Landscape Ecology **19:** 291–309, 2004.

Research article

Disturbance and landscape dynamics in the Chequamegon National Forest Wisconsin, USA, from 1972 to 2001

Mary K. Bresee[1,*], Jim Le Moine[1], Stephen Mather[1], Kimberley D. Brosofske[2], Jiquan Chen[1], Thomas R. Crow[3] and John Rademacher[1]

[1]*Department of Earth, Ecological, and Environmental Science, University of Toledo, OH 43606, USA;* [2]*School of Forest Resources and Environmental Science, Michigan Technological University, Houghton, MI 49931, USA;* [3]*Forestry Sciences Lab, USDA Forest Service, Grand Rapids, MN 55744, USA;* [*]*Author for correspondence (e-mail: bres9575@hotmail.com)*

Received 25 September 2002; accepted in revised form 12 May 2003

Key words: Disturbance, Forest management, Fragmentation, GIS, Landscape dynamics, Landscape structure, Landsat MSS, Roads, TM and ETM+, Wisconsin

Abstract

Land uses, especially harvesting and road building, are considered to be the primary cause of forest fragmentation in many parts of the world. To test this perception, we (1) quantified changes and rates of change in vegetative composition and structure within the Washburn Ranger District in northern Wisconsin using Landsat images, (2) examined changes in landscape structure, (3) assessed changes within the area of road influence (ARI), and (4) investigated changes in landscape composition and structure within the context of forest management activities. Our landscape classifications included six dominant cover types: mixed hardwood (MH), jack pine (JP), red pine (RP), mixed hardwood/conifer (MHC), non-forested bare ground (NFBG), and regenerating forest or shrub (RFS). Increases in NFBG and RFS, by 196% and 28% respectively, reflect expansion of the pine-barrens. Windthrow in the mature hardwoods during the late 1970s and jack pine budworm outbreaks during the mid-1990s correlated with decreases in those classes over the corresponding intervals. A 69% decrease in mean patch size and a 60% increase in edge density reflect increased fragmentation. An inverse relationship existed between the compositional trends of forested (excluding JP) cover types and RFS and NFBG cover types. ARI covered 8% of the landscape affecting species composition within the MH, RFS, and NFBG. Results from this study are key in assessing the links between management activities and ecological consequences and thereby facilitate adaptive management.

Introduction

Forest changes in the Great Lakes Region have been ubiquitous during the last 150 years. In most landscapes within the region, land-use is a major determinant of composition, spatial patterns and function at the species, community and landscape levels (Pan et al. 2001). Timber harvesting and associated silvicultural practices (e.g., clearcutting, thinning, plantation forestry, and prescribed burning), and road building alter spatial and successional patterns as well as the underlying processes and dynamics of forest ecosystems. Consequently, forest management practices can lead to landscape fragmentation (Euskirchen 2001). For example, edge effects are the most significant consequence of natural and anthropogenic forest fragmentation (Franklin and Forman 1987; Chen et al. 1999). The area of edge influence (AEI) within a forested landscape is an important determinant of forest connectivity, heterogeneity, and species composition (Euskirchen et al. 2001). Roads are a major contributor to edge creation in a landscape. The total area of a landscape subjected to significant road effects is known as the area of road influence (ARI). ARI is the only AEI considered in this study and is often defined by altered microclimate (e.g., soil temperature, moisture, etc., Saunders et al. 2001), water infiltration, canopy closure and weed dispersal, all of which impact species composition within the surrounding areas (Watkin et al. 2003). Clearly, understanding changes of each landscape elements is the key for synthesizing the overall function of a landscape.

Documenting and calculating the rates of landscape change, using satellite imagery, is a useful approach for exploring potential underlying mechanisms affecting ecological processes and ultimately human impact on forested landscapes (Cohen et al. 2002; Zheng et al. 1997; Walsh et al. 1998). Satellite imagery, verified with field data, can provide a broadscale spatial characterization of landscape change needed to assess habitat pattern and composition. Accurate mapping of cover types or successional stages can facilitate the assessment of environmental conditions, changes, and trends of a landscape. Ultimately, documented rates and patterns of change in forested ecosystems can be applied in decision and planning processes (Cushman et al. 2000; Cohen et al. 1995).

Using a combination of remote sensing imagery and geographical information system (GIS) techniques, we evaluated the overall pattern, composition, and structure of a managed landscape in the Washburn District of the Chequamegon National Forest (CNF) over a 29-year period, focusing on the six predominant forest cover types. Additionally, we examined relationships between landscape structure and major management regimes and disturbance events. The landscape has been the subject of intensive ongoing research since 1994. The study was initiated to evaluate the consequences of broad-scale forest management regimes on various ecological parameters. An extensive database, including information on microclimate, vegetation, decomposition, soil respiration, and other variables, has been compiled with investigations focusing on effects of landscape structure, pattern-process relationships, and tradeoffs among various management approaches within this landscape (Saunders et al. 1998, 1999, 2002; Brosofske et al. 1999; Brosofske et al. 2001; Chen et al. 1999; Euskirchen et al. 2001; Zheng and Chen 2001, Watkins et al. 2003). The previous field studies provided valuable information concerning the current landscape but lack the long-term perspective that could provide a more complete context for adaptive management of the landscape.

The specific objectives of this study were to: (1) quantify the changes and rates of change between 1972 and 2001 in vegetative composition across a landscape managed for multiple uses within the CNF in northern Wisconsin using Landsat images, (2) examine the changes in landscape structure (e.g., fragmentation, total edge length, diversity, and evenness), (3) assess compositional changes, over time, within the area of road influence (ARI, delineated as the total area lying within 57 m of all existing road, was chosen based on the resolution of the images.), and (4) investigate and discuss the changes observed in the landscape composition, pattern and structure over time in the context of forest management activities and other disturbances.

Methods

Study location

The study area was located in the Washburn Ranger District of CNF in northern Wisconsin (46°30'-46°45' N, 91°02'-91°22' W). The study location has been exposed to extensive disturbance, including fire and timber harvesting, throughout the past century, which has created a variety of landscape mosaics that are representative of many forested landscapes throughout the eastern United States. The northern portion of CNF covers approximately 49,818 ha. Due to image limitations, we classified approximately 39,381 ha (79%) of the total study area (Figure 1).

Soils consist of deep (30-90 m) loamy, glacial outwash sands, classified as Psamments and Orthods. The underlying Precambrian shield bedrock is characterized by basalts, lithic conglomerates, sandstones, shale and feldspathic to quartzose sandstones. The topography is typically flat to gently rolling while landforms consist of level terraces and pitted outwash plains, with elevations ranging from 232-459 m. The study area lies within Section 212K (Western Superior Section of the Laurentian Mixed Forest Province, Humid Temperate Domain) and Subsection 212Ka of the National Hierarchical Framework of Ecological Units (Bailey 1995; McNab and Avers 1994). There are two LTA's (Land Type Associations) 212Ka 04 and 212Ka 07, and the landform is Bayfield Barrens (Subsection X.1). Climate is typified by characteris-

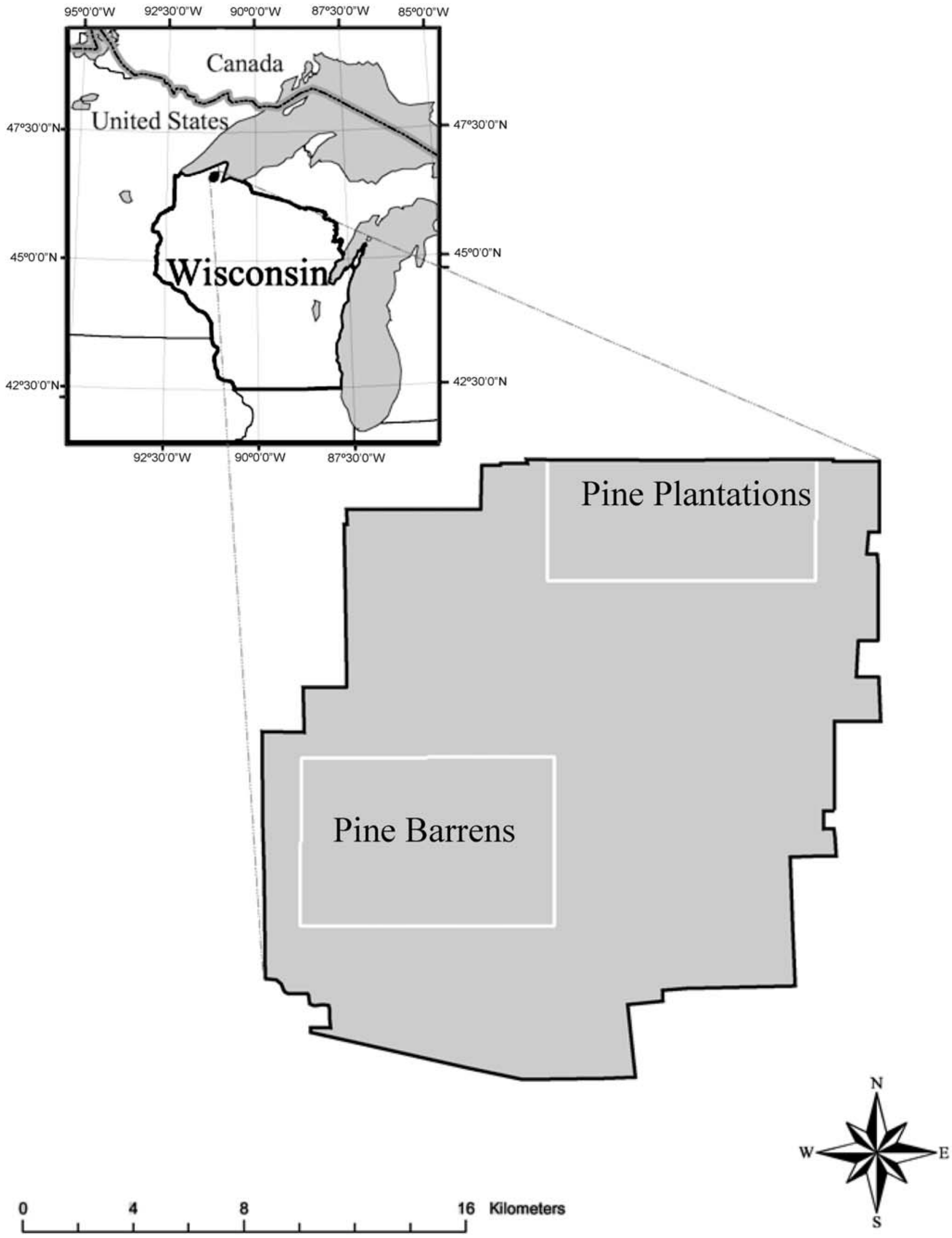

Figure 1. Site location map of the Washburn Ranger District of the Chequamegon National Forest. The study area is a subset within the ranger district limited by Landsat image availability. Smaller regions depict the pine plantations of the northeast and the pine barrens of the southwest, depicted as smaller boxes within the study area.

tically short, hot summers and long, cold winters. The growing season lasts 120-140 days, and average yearly precipitation ranges between 66-70 cm rain and 106-150 cm snowfall (Albert 1995).

The original land survey of Wisconsin (1832-1866) indicated a mixture of red, white and jack pine dominated the landscape. However, management has heavily altered the species composition in this region; currently, aspen/birch stands tend to be the principal species in the landscape. We focused our analysis on the six dominant cover types in the study area as identified by total area. These cover types included:

Table 1. Brief descriptions of the six predominant cover types in the Washburn Ranger District in Chequamegon National Forest.

Cover Type	Description
Mixed Hardwood (MH)	A closed canopy Red Pine (RP) The red pine (mixture of hardwoods in varying proportions; species include red maple (*Acer rubrum* L.), sugar maple (*Acer saccharum* Marsh.), red oak (*Quercus rubra* L.), paper birch (*Betula papyrifera* Marsh), quaking aspen and big-toothed aspen (*Populus tremuloides* Michx. and *P. grandidentata* Michx.)
Red Pine (RP)	The red pine (*Pinus resinosa* Ait.) primary source of commercial timber; typically large homogeneous plantationslocated in the northeastern portion of the Washburn District
Jack Pine (JP)	Jack pine (P. *banksiana* Lamb.) typically occurs as small isolated patches distributed randomly throughout thelandscape, and as an edge species of various cover types.
Mixed Hardwood/Conifer (MHC)	A closed-canopy mixture of hardwoods, which varies by site. Includes a combination of conifer listed above as well as white pine (*P. strobes* L.), black spruce (*Picea mariana* Mill.), white spruce (*Picea glauca* Moench), eastern hemlock (*Tsuga canadensis* L.), white cedar (*Thuja occidentalis* L.) and mixed hardwood.
Regenerating Forest or Shrub (RFS)	Generally occurs within approximately 15 years after a disturbance and is typified by a canopy that has notclosed; includes both regenerating forests and shrub/scrub areas that are maintained in an open condition by fire or other disturbance (e.g., pine barrens).
Non-Forested Bare Ground (NFBG)	Occurs directly following disturbance (management or natural) and is characterized by limited vegetation
Other	Water

mixed hardwood (MH), jack pine (JP), red pine (RP), mixed hardwood/conifer (MHC), non-forested bare ground (NFBG), and regenerating forest or shrub (RFS) (Table 1). We placed forest stands into the MH, JP, RP, and MHC classifications based on composition and the presence of an identifiably closed canopy. Openings smaller than 57m×57m (i.e., one pixel) were assumed to be canopy gaps and were incorporated into the appropriate forested cover type. Young, regenerating forests and other vegetation in which an open canopy predominated were placed into either the NFBG or RFS class, depending on whether the vegetative cover was limited (NFBG) or abundant (RFS). The two open-canopy cover types were typically areas in which disturbance (e.g., harvesting, fire, windthrow) had recently occurred.

Past and present management

Historically, the structure of the study landscape was shaped primarily by fire originating from both natural and anthropogenic causes (i.e., Native Americans and European settlers, Heinselman 1981). Windthrow, insect damage, and farming were additional disturbances structuring the landscape. During the last century, however, fire suppression, timber harvesting and other silvicultural practices (e.g., plowing and planting of the pine-barrens in the 1930s by the Civilian Conservation Corps. CCC), and corridor construction (e.g., roads, powerlines) have severely altered the structure of the landscape.

Several major periods of forest management can be identified. First, the emphasis of forest management prior to the mid-1980s was directed primarily at maximizing timber output, largely through clearcutting and plantation forestry. In the 1930s − 1940s, the CCC plowed and planted much of the study area with red and jack pine; these plantations became harvestable in the 1970s, further promoting timber harvests.

The current forest management plan for the CNF originated in 1986 and is being revised. The 1986 plan emphasizes management for early- and mid-successional species for timber and wildlife through use of silvicultural techniques such as clearcut, shelterwood, and seed tree harvests (Saunders et al. 1998). The plan's focus on stand- and species-based management has indirectly and directly promoted landscape fragmentation through prescriptions calling for small, dispersed patches. Clearcutting typically is restricted to less than 16 ha except in the pine-barrens and during salvage situations where the limit is 405 ha.

During the early 1990s, recognition of the inadequacies of traditional, stand-level forest management led to an increasing emphasis on broad-scale issues and management of the whole ecosystem or landscape (Overbay 1992), and ecosystem management became the guiding paradigm for management of federal forest lands. CNF outlined its ten "desired future condition" management areas with respect to the broad-scale pattern in addition to the vegetative composition desired (USDA 1993) and proceeded to implement the plan. The current Forest Plan revisions incorporate this broad-scale approach and emphasize the need to address additional aspects of the forest ecosystem, including non-game wildlife and plants, forest health, ecosystem restoration, and social functions, all of which can be strongly related to landscape pattern and dynamics.

The jack pine and red pine stands are predominately plantations with natural stands interspersed. Jack pine stands are clearcut on a 40 to 70 year interval, and red pine stands are thinned every 7 to 15 years until they approach 100 to 150 years old, at which time they are harvested under a shelterwood regime. Prior to 1986, harvesting of RP occurred when trees reached 40 to 60 years old or when the trees were of merchantable size. Presently, MH and MHC are separated into three classes – even-aged short rotation, even-age long rotation, and uneven-aged long rotation (USDA 1986).

Specific restoration goals of the pine-barrens within the current forest management plan are to maintain natural and plantation jack pine mixed with large temporary openings through harvesting and prescribed fire. Harvest rotations will be shortened to a 35 to 40 year rotation to maintain the desired composition and structure.

Landsat images and preprocessing

We acquired six Landsat images of the study area, between 1972 and 2001, from the United States Geological Survey (USGS). The earlier images (1972, 1978, and 1982) were from the Multispectral Scanner (MSS), images 1987 and 1992 were from the Thematic Mapper (TM) sensors, and the 2001 image was from Enhanced Thematic Mapper plus (ETM +). All images were obtained during the summer months of May, June, July and August. Pixel size was 30×30 m for the Landsat TM images and 57×57 m for the Landsat MSS images, resulting in inconsistencies in area. These discrepancies were adjusted in the TM and ETM+ images (1987, 1992 and 2001) using the resampling function (after classification) in ERDAS 8.5 (Earth Resource Data Analysis System) for future pixel-by-pixel comparison. All images corresponded well spatially with GIS data layers of roads and lakes. Inter-image spatial error was also low, with a root mean square error (RMSE) of 48 m between the 1972 and 2001 images.

To ensure a degree of consistency among all images, the following procedures were taken: (1) geo-rectifying all images to World Geodetic System 84 (WGS_84) using ArcGIS 8.1 (ESRI- Environmental Systems Research Institute) and ERDAS 8.5 software, and (2) masking of clouds, cloud shadows and lakes in all images using the spatial analysis tool in ArcGIS 8.1. Lakes were masked in order to focus on terrestrial vegetation. The lakes mask for all years was acquired from the U.S. Census Tiger files, which corresponded well with the locations of lakes in the imagery.

All images required adjustment due to the effects of atmospheric conditions and resolution differences between years. The resolution in earlier Landsat MSS images inhibited the classification process; further enhancement of these images was necessary before classifications could take place. Using ERDAS 8.5 pixel definition, we cleared the images using a crisping function, which ultimately increased the separation among all cover types. The crisping function was not used on the Landsat TM and ETM+ images due to the higher resolution of the pixel values; instead the haze reduction process in ERDAS 8.5 was used.

Classification

Initially we performed a supervised maximum likelihood classification on all images using spectral training sets, based primarily on known stands of each cover type (Duda and Hart 1973; Schowengerdt 1983). Utilizing the threshold tool in ERDAS 8.5, histograms were created for each cover type: MH, PINE (RP and JP), MHC, RFS and NFBG. Cover types were separated by determining the spectral distance between the class signatures. Distinguishing between forested and non-forested cover types was a relatively straightforward process, but separating the forested cover types was more difficult.

Large spectral variability within the MHC class was caused primarily by variability in species composition. In order to account for this, two distinct spectral classes, one representing stands of predominantly hardwoods with conifers throughout and the other representing coniferous stands with hardwoods mixed throughout, were created and later combined. The pine classes (RP and JP) were separated by masking out all other vegetation classes to enhance differences between the two classes and thereby reduce misclassifications. Once the conifer classes were separated from the other classes, new spectral training sets were created from known JP and RP stands. Differentiation between JP and RP was based on stand

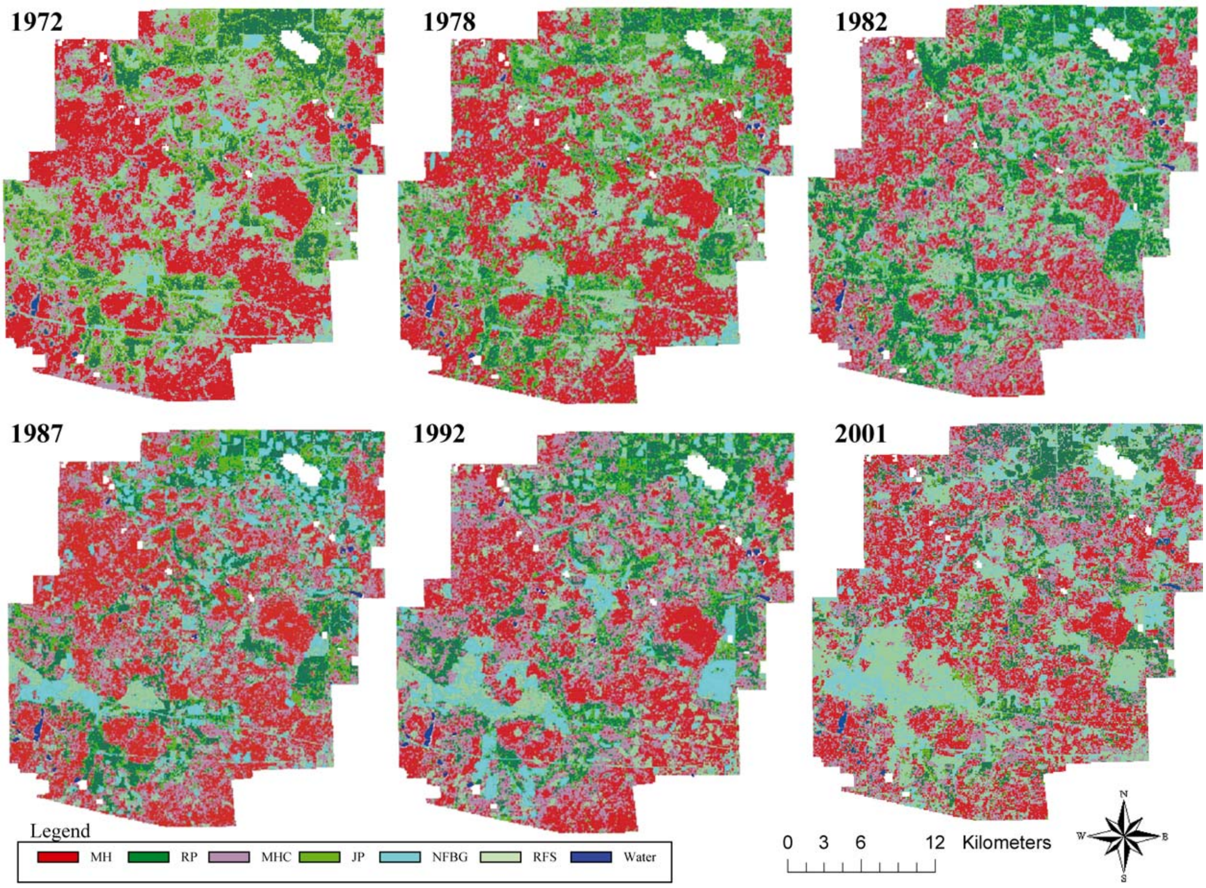

Figure 2. Six classified images (1972-2001) of the landscape. Cover types include mixed hardwood (MH), jack pine (JP), red pine (RP), mixed hardwood/conifer (MHC), regenerating forest or shrub (RFS), and non-forested bare ground (NFBG).

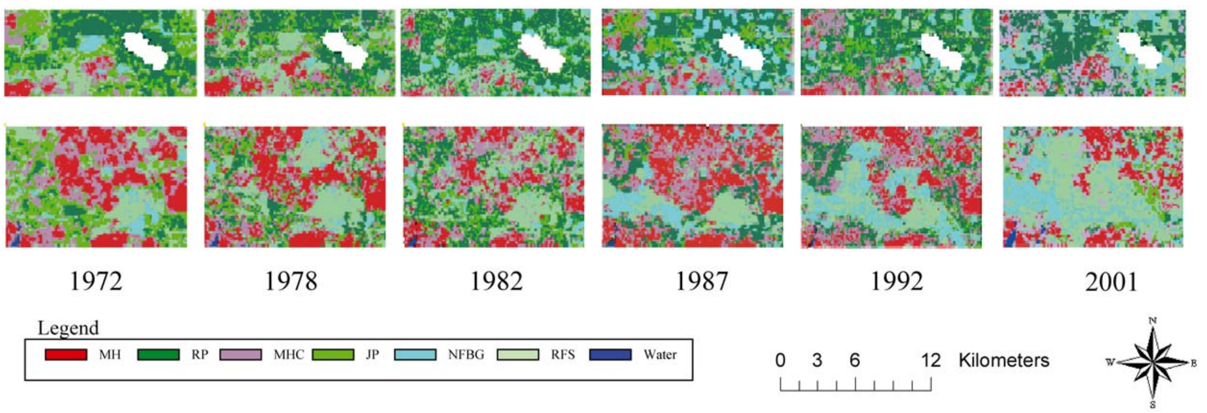

Figure 3. The twelve subsets from the initial classified images. The upper row of images depict the pine plantations in the northeast portion while the lower row depict the pine barren subset in the southwest portion of Chequamegon National Forest.

data obtained from the Forest Service and from field observations. Similar to the initial supervised classification, the knowledge-based supervised classification employed a maximum likelihood algorithm (Figure 2).

Landscape change detection

A post-classification analysis based on the supervised classifications was used to detect landscape change. We investigated changes using several measures of landscape structure composition which were calculated and compared from image to image. To allow pixel-by-pixel comparison for all years, we resampled pixels in the 1987, 1992, and 2001 images to a 57 m resolution to match the earlier images. For each of the resulting images, we calculated the patch characteristics edge density (m ha^{-1}), mean patch size (ha), median patch size (ha), maximum patch size (ha), coefficient of variation, clumpiness (%), total patch number, and total edge length (m), using FRAGSTATS 3.3 (McGarigal 2002). Landscape metrics, including Shannon diversity (0-1) and evenness indices, Simpson's evenness (0-1) index and contagion (0-100), were also calculated. Total area of each cover type (ha) and the mean edge length were calculated using a combination of ERDAS 8.5, ArcINFO 8.2 (ESRI) and Splus 6.0 (Insightful). Mean edge length was defined as the mean edge per patch for a given cover type.

Finally, we applied a 57 m (e.g., 1 pixel length on each side) buffer to all roads and calculated the area of each cover type falling within the ARI. All areas not influenced by roads are considered interior patches. The road system used in the buffering process of all images consisted of the current permanent road matrix; information on road system changes was collected from the Forest Service engineering department, and it was concluded that main road systems did not change within the 29-year period. Since temporary haulage roads and landings were not included in this study, estimates of ARI may be conservative.

Initially we intended to do a pixel-by-pixel composition change detection. Due to sensor pixel size and orbital path differences, we considered this analysis to be problematic at the pixel scale. Instead, two subset regions were used for illustrative purposes depicting land cover change (1) in what is known as the pine-barrens region and (2) the northern plantations of RP and JP (Figure 3).

Calculating changes using multi-year classifications typically suffer from compounding errors from changes in radiance and radiance values (Mas 1999). Of the six change detection techniques, Mas (1999) found that post-classification comparison had the highest accuracy. Within our study, error was considered, as classification errors are larger than the rates of change observed in each patch type and were noted with an asterisk (*) in all pertinent tables. It was assumed that changes smaller then the classification error meant limited change in cover type area had occurred. To assure the robustness of all classifications, an accuracy assessment was performed both before and after modifications were made.

Accuracy assessment

The accuracy assessments were a pixel-by-pixel comparison of all images using a set of 146 ground-truthed data points. In addition to the data points, seven training sets were created using ground data from geographically distinct and identifiable locations, such as road intersections. Information collected for each training set included vegetation type, approximate area, and age, allowing further use in all classified images.

The large temporal period covered by this project required the use of multiple data and information sources to assess the accuracy of each image. For verification of accuracy in earlier images, a combination of the Forest Service data, training sets, and aerial photographs were used to supplement the known ground control points. The data from the Forest Service included stand information spanning from 1977 to the present, enabling us to determine years of origin as well as vegetation types of stands across the entire landscape. Finally, all the assessment points were compared to aerial photographs taken in years 1973, 1979, 1988 and 1993 (leaf-off), assisting in the differentiation of classes in older images. A preliminary accuracy assessment was performed on all images prior to resampling of pixels and filtering to determine the relative accuracy of the images. The final assessment was completed following both the resampling and filtering processes. Results were compared to determine the relative error associated with the manipulation of classified images. Because the differences were minor, we proceeded with the examination of composition, patch size, and ARI. The overall accuracy for all images and cover types was 80.1% (Table 2).

Table 2. Error matrix for all cover types and years based on 146 assessment points. Includes producer and user accuracy of each cover type for each image, total accuracy and weighted total of all images and cover types.

Year	MH	RP	MHC	JP	RFS	NFBG	Total	Weighted
1972								
Producer error	82.6	72.7	81.3	65.8	93.5	93.8	81.6	81.2
User error	86.4	69.6	61.9	80.6	90.6	88.2	79.6	78.8
1978								
Producer error	81.8	94.7	50.0	48.0	100.0	50.0	70.1	75.5
User error	77.1	69.2	50.0	85.7	89.3	100.0	78.7	73.5
1982								
Producer error	75.0	83.3	73.3	56.7	97.2	88.9	79.1	79.5
User error	91.3	65.8	61.1	77.3	92.1	88.9	79.4	78.1
1987								
Producer error	90.0	75.0	88.0	52.6	92.3	92.9	81.8	85.8
User error	90.0	62.8	73.3	74.1	92.3	100.0	82.1	80.9
1992								
Producer error	65.2	81.3	96.7	55.6	100.0	100.0	85.8	84.4
User error	93.8	63.4	70.7	87.0	100.0	100.0	83.1	83.7
2001								
Producer error	64.7	73.1	81.3	69.0	83.3	100.0	81.8	77.7
User error	73.3	65.5	86.7	87.0	93.8	84.6	77.6	82.3
Total	80.9	73.0	72.9	69.9	93.7	90.6	80.1	80.1

Results

Landscape change

Area

JP experienced the most extensive change throughout the 29-year interval, decreasing in area from 6027 ha in 1972 to 2070 ha in 2001 (Table 3). The majority of the loss to the JP cover type (− 4017 ha, or ~ 67.0% of the cover type) occurred during the first 15 years (1972–1987) (Figure 4). Conversely, the NFBG had a fairly consistent increasing trend between 1972 and 2001. The NFBG class decreased between 1972 and 1978 by 672 ha (~50.0%) followed by a net increase of 3298 ha over the remaining 23 years, almost tripling its original size.

There were no clear trends associated with the RFS, MH, and MHC cover types, primarily because of large fluctuations. RFS increased steadily (+1539 ha), in area between 1972 and 1982, but between 1982 and 1987 there was a reduction in area (− 3306 ha) representing approximately a 38.0% decrease in overall coverage. Following this decrease, the RFS gained 3834 ha during the last 14 years. MH and MHC were the most dominant cover types over the past 29 years, averaging approximately 28.0% and 23.0% respectively, of the total landscape (Table 3). The largest reduction in MH area (4164 ha) occurred between 1978 and 1982, whereas the principal increase of 2525 ha occurred between 1982 and 1987 (Table 4). MH experienced a 13.0% net reduction in total area, while MHC experienced a minor net decrease (0.09%). The RP increased in area by approximately 22.0% (3540 to 4311 ha) overall. Although there were not large fluctuations within the RP cover type, there were also no obvious trends.

JP had the fastest rate of change throughout the study period, losing approximately 136.4 ha yr^{-1} throughout the entire period (1.2% per year, Table 4). The initial 15 years (1972 to 1987) experienced the greatest rates of change, with losses ranging between 344.7 ha and 412.2 ha yr^{-1}. The overall rates of change for both the NFBG and RFS cover types were very similar, approximately 0.8% change per year. Although the rates of change were similar, only the NFBG had a clear increasing trend throughout the entire period. The RFS fluctuated within the first 10 years, but showed a gradual increase throughout the remaining time.

The rates of area change for the MH, MHC, and RP fluctuated throughout the entire period with little indication of increasing or decreasing trends. The MH was typically the most dynamic cover type, with large fluctuations throughout the 29-year interval. MH generally experienced the highest rates of change

Table 3. Total area (ha), mean patch size (MPS, ha), standard deviation of MPS, maximum patch size (MXPS, ha), coefficient of variation (COVAR), clumpiness (CMP, %) and the frequency of distribution for total area (*f*TA) and area of road influence (*f* ARI) for all cover types in each landscape image.

year	Cover Type	TA (ha)	MPS (ha)	Std MPS	MXPS (ha)	CO VAR	CMP	*f* TA	*f* ARI
1972									
	MH	11554	16.9	128.5	2191	756	0.72	0.29	0.15
	RP	3540	4.7	32.7	1383	703	0.61	0.09	0.08
	MHC	9732	5.3	31.2	861	584	0.44	0.25	0.22
	MHC	9732	5.3	31.2	861	584	0.44	0.25	0.22
	JP	6027	3.2	11.7	22	369	0.44	0.15	0.16
	RFS	7195	3.6	18.0	385	502	0.53	0.18	0.31
	NFBG	1341	2.4	6.8	41	275	0.61	0.03	0.08
Total	39390								
1978									
	MH	12736	11.4	115.4	2625	1011	0.65	0.32	0.16
	RP	4505	3.5	25.2	1383	719	0.61	0.11	0.11
	MHC	9405	2.8	7.7	145	271	0.33	0.24	0.25
	JP	3958	2.4	4.7	22	198	0.45	0.10	0.09
	RFS	8129	4.8	25.1	530	518	0.61	0.21	0.34
Total	39402								
1982									
	MH	8572	4.1	35.6	1383	868	0.52	0.22	0.16
	RP	6449	4.5	27.5	536	615	0.49	0.16	0.19
	MHC	10038	5.4	29.1	655	534	0.43	0.25	0.21
	JP	4071	1.4	3.4	22	244	0.27	0.10	0.11
	RFS	8734	3.6	19.9	554	548	0.48	0.22	0.27
	NFBG	1537	1.9	4.5	54	236	0.54	0.04	0.06
Total	39401								
1987									
	MH	11097	6.0	70.4	2443	1176	0.55	0.28	0.17
	RP	5543	3.9	26.3	617	678	0.58	0.14	0.19
	MHC	12678	4.1	22.0	567	540	0.36	0.32	0.30
	JP	2010	1.0	3.3	54	321	0.31	0.05	0.09
	RFS	5428	2.0	8.9	23	456	0.43	0.14	0.20
	NFBG	2596	2.8	9.4	1383	329	0.59	0.07	0.05
Total	39351								
1992									
	MH	10917	4.9	56.0	1727	1146	0.59	0.28	0.19
	RP	5981	4.0	42.7	1383	1056	0.55	0.15	0.19
	MHC	11291	4.5	38.8	899	701	0.46	0.29	0.26
	JP	1872	0.8	1.7	54	212	0.22	0.05	0.10
	RFS	6475	2.4	15.0	618	621	0.47	0.16	0.17
	NFBG	2857	3.6	17.8	356	488	0.64	0.07	0.10
Total	39392								
2001									
	MH	10094	5.3	47.1	1383	896	0.51	0.26	0.15
	RP	4311	2.3	16.6	649	707	0.47	0.11	0.11
	MHC	9647	3.0	17.1	464	573	0.34	0.24	0.20
	JP	2070	0.8	1.5	42	193	0.22	0.05	0.05
	RFS	9262	3.2	34.7	1722	1091	0.45	0.24	0.32
	NFBG	3967	1.9	11.7	458	604	0.45	0.10	0.16
Total	39350								

within the first 14 years, ranging between -12.5% and 4.0% change per year.

An inverse relationship existed between the rate of change of the relatively open-canopy cover types (RFS and NFBG) and that of the closed-canopy for-

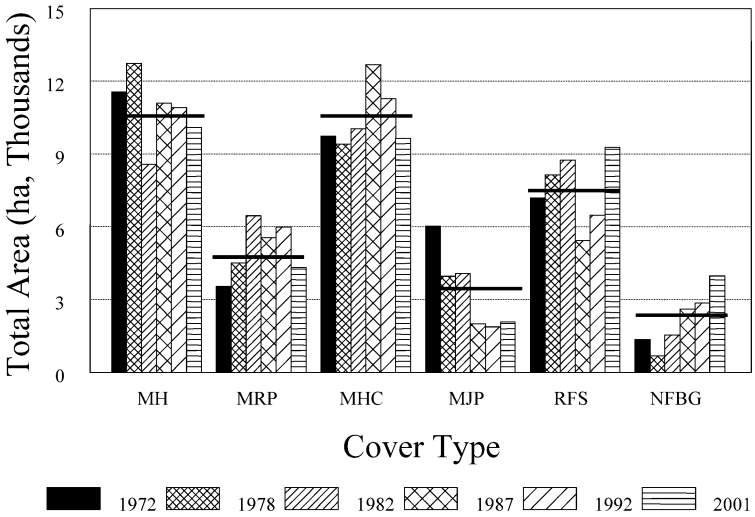

Figure 4. Total area (ha) of each cover type class within Chequamegon National Forest throughout the study period (1972-2001). Line across values indicates the mean area of each patch type.

ests (RP, JP, MHC, MH). For example, the RFS and NFBG combined have experienced positive rates of change (e.g., 1987 to 1992 ~ 261.7 ha yr^{-1}, and 1992 to 2001 ~ 433.0 ha yr^{-1}), whereas the rest of the cover types combined have undergone negative rates of area change (e.g., 1987 to 1992 ~ 253.4 ha yr^{-1} and 1992 to 2001 ~ 437.5 ha yr^{-1}).

Patch characteristics

All cover types, excluding JP, decreased in mean patch size (ha) and clumpiness (%) while increasing in patch abundance throughout the entire time interval (Table 3). Additionally, the maximum patch size (ha), coefficient of variation and standard deviation of the mean patch size decreased in the forested cover types, conversely non-forested cover types increased. Mean patch size within the JP cover type showed an almost continual decline between 1972 and 2001 (3.2 to 0.8 ha, ~75.0% reduction) with the greatest decrease occurring between 1972 and 1978 (Figure 5b). Conversely, patch abundance within the JP cover type increased by 2691 patches between 1972 and 1982, in the ensuing 19 years it decreased by 1446 patches. Clumpiness and the standard deviation of the JP mean patch size decreased between 1972 and 2001 (~22.0% and 87.2%, respectively). Mean patch size within the NFBG cover type showed a clear increasing trend, from 1972 to 1992, overall, it increased by approximately 33.3%. Within the same time span, patch abundance and maximum patch size increased by approximately 634 to 1120 patches and 41 ha to 356 ha respectively. Clumpiness within the NFBG and RFS decreased throughout the entire time interval by approximately 16.0% and 9.0%, respectively. Mean patch size within the RFS tended to show trends opposite to the other forest cover types. For example, RFS mean patch size increased by 33.3% in 1978, whereas patch size of all other forest cover types decreased (e.g., MH and JP patch size decreased between 1972 and 1978 ~ 31.4%).

The mean patch size within the MH decreased by 11.6 ha between 1972 and 2001. MH had the largest mean patch size of all cover types during 1972 and 1978 (16.9 ha and 11.4 ha, respectively), ranging between 7.0 and 2.4 times larger than all other cover types. Between 1978 and 1982, mean patch size of the MH cover type experienced a 65.0% reduction, decreasing from 11.4 ha to 4.1 ha. Throughout the ensuing 18 years, mean patch size fluctuated between 5.3 ha (2001) and 4.9 ha (1992). Clumpiness and maximum patch size decreased (~21.0% and 36.9%, respectively) whereas patch abundance within the MH cover type increased steadily throughout the last 29 years, from 1284 to 3707 patches a 66.3% increase. Mean patch size and patch abundance within the MHC and RP did not follow any particular tends throughout the study. Unlike the other measurements of central tendency median patch size for all cover types showed no distinct trend, varying between 0.3 ha and 0.7 ha throughout the entire 29 years.

Landscape evenness, contagion and diversity in all images did not experience large changes between

Table 4. Area change (ha), mean and percent rate of area change (ha yr^{-1}and %.yr-1respectively) of each cover type between classification years. Asterisks (*) indicate classification errors that are larger than the rates of change observed in each patch type

Cover Type	ha	ha/yr	%/yr
Change: 1972 to 1978			
MH	1181.7	196.9	3.2
RP	965.0	160.8	2.6
MHC	− 327.5	− 54.6	− 0.9
JP	− 2068.3	− 344.7	− 5.6
RFS	933.8	155.6	2.5
NFBG	− 671.9	− 112.0	− 1.8
Change: 1978 to 1982			
MH	− 4163.6	− 1040.9	− 12.5
RP	1943.9	486.0	5.8
MHC	633.2	158.3	1.9
JP	112.7	28.2	0.3
RFS	604.6	151.2	1.8
NFBG	867.8	217.0	2.6
Change: 1982 to 1987			
MH	2524.5	504.9	4.0
RP	− 906.5	− 181.3	− 1.5
MHC	2639.8	528.0	4.2
JP	− 2060.8	− 412.2	− 3.3
RFS	− 3305.5	− 661.1	− 5.3
NFBG	1058.5	211.7	1.7
Change: 1987 to 1992			
MH	− 179.7	− 35.9	− 1.0
RP	438.0	87.6	2.5
MHC	− 1387.0	− 277.4	− 8.0
JP	− 138.4	− 27.7	− 0.8
RFS	1046.8	209.4	6.1
NFBG	261.5	52.3	1.5
Change: 1992 to 2001			
MH	− 823.0	− 91.4	− 1.1
RP	− 1669.8	− 185.5	− 2.3
MHC	− 1643.8	− 182.6	− 2.2
JP	198.2	22.0	0.3
RFS	2787.0	309.7	3.8
NFBG	1109.6	123.3	1.5
Change: Total			
MH	− 1460.1	− 50.3	− 0.5
RP	770.5	26.6	0.2
MHC	− 85.2	− 2.9	0.0
JP	− 3956.6	− 136.4	− 1.2
RFS	2066.7	71.3	0.6
NFBG	2625.5	90.5	0.8

1972 and 2001. Between 1972 and 1982, Shannon and Simpson's evenness increased from 0.90 to 0.93 and from 0.90 to 0.95, respectively (Table 5). These indices exhibited an inverse trend throughout the remaining 19 years, decreasing from 0.93 to 0.86 and 0.95 to 0.93 respectively. Contagion values were low (0-100) with a maximum value of 26.6 (1972). The largest decrease occurred between 1972 and 1982 (approximately 26.6 to 21.0, respectively); conversely, between 1982 and 2001 the contagion increased (21.0 to 23.5 respectively). Simpson's diversity increased between 1972 and 1982 (1.60 to 1.67, respectively). Subsequent to a rapid decrease between 1982 and 1987 the diversity index increased between 1987 and 2001 (1.60 to 1.67, respectively).

Edges

The trends of mean edge length (m) within all cover types were similar to changes within mean patch size (Figure 5b, Figure 5c). Mean and total edge length within the JP steadily decreased between 1972 and 2001 (Figure 5c, Table 6), indicating the area of this cover type was decreasing throughout the landscape (Figure 4). The total edge length of NFBG, RFS, MH, and RP increased (290.1%, 43.6%, 54.0% and 63.1%, respectively), while mean edge lengths decreased (18.6%, 16.5%, 47.0% and 29.5%, respectively) between 1972 and 2001 (Table 6). There were no clear trends for NFS and NFBG because of large fluctuations. The mean patch size of RP and MH followed a decreasing trend from 1972 to 2001. During the 1970s, mean edge length within the MH was substantial (1390 m in 1972 and 1275m in 1978) and total edge length was small (1757367 m in 1972 and 2283363 in 1978) relative to other classes, indicating that patch size during this time was relatively large (Figure 5b).

Edge density (Figure 5d) increased within each cover type area except MHC, with the most noticeable difference occurring in the MH, RP and JP cover types. The edge density within these cover types indicates large increases in edge along with corresponding decreases in patch size, throughout the study period; in some cases (e.g., JP and MH) the ratio nearly doubled in size – an indication of elevated fragmentation.

Roads

The ARI comprised approximately 8.0% of the total landscape throughout all years. JP and NFBG were the only two cover types to show any type of trend, with JP decreasing and NFBG increasing in area within the ARI. MH fluctuated least throughout the study period, with the ARI varying between 15.0% and 19.0%. The remaining cover types fluctuated greatly, not showing any trend or pattern. However, we still found an inverse relationship when comparing RFS and NFBG with the remaining cover types. Between 1992 and 2001, for example, the area of RFS

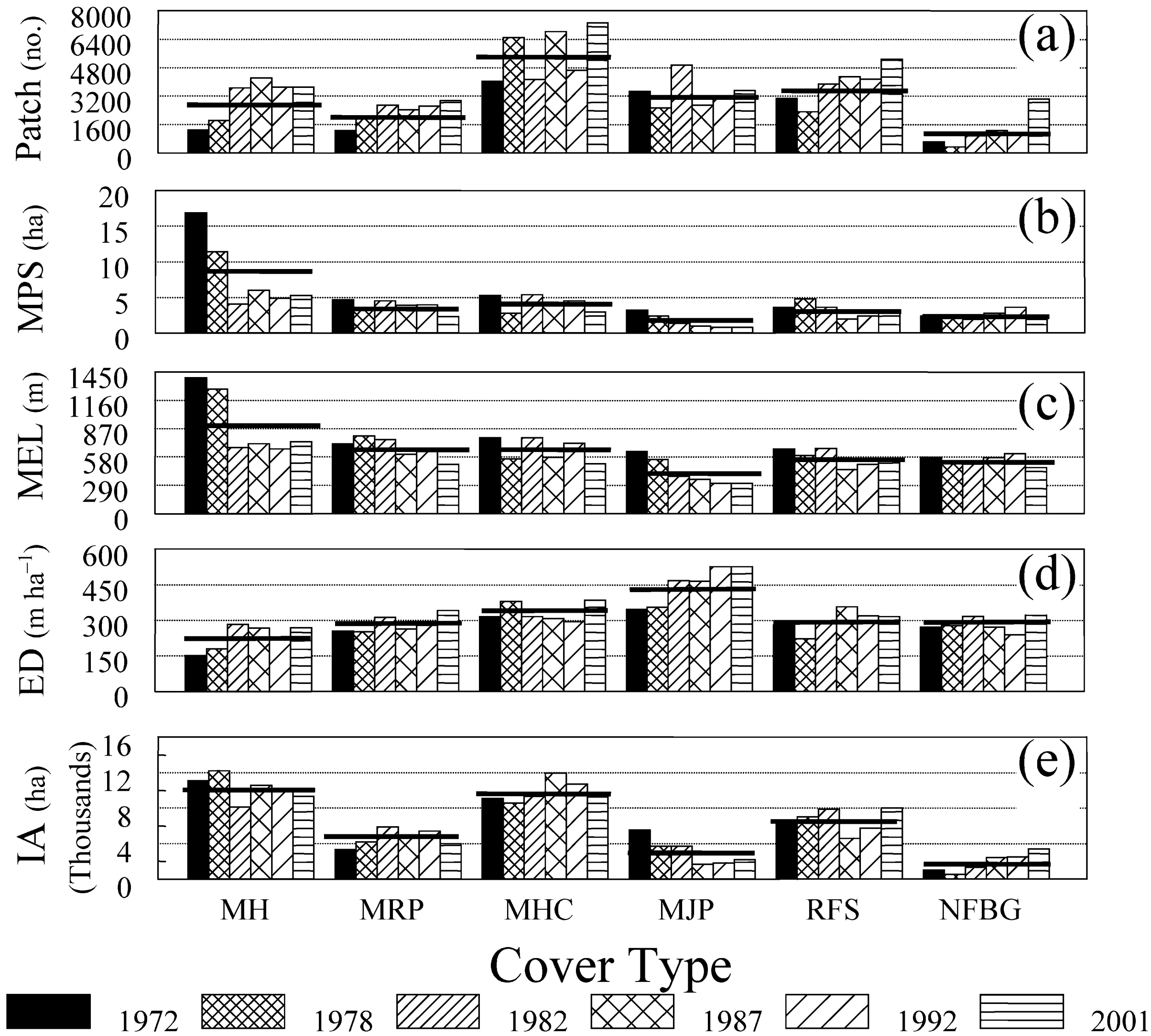

Figure 5. Patch structure and fragmentation within Chequamegon a) patch abundance (patch no.), b) mean patch size (MPS, ha) c) mean edge length (MEL, m), d) edge density (ED, m ha^{-1}), and e) interior area (IA, ha) of all cover types throughout the 29-year interval. Line across values indicates the overall average of each patch type (e.g., patch abundance, patch size, edge length, edge density and interior area).

Table 5. Landscape indices by year contagion (CONT, 0-100), Simpson's diversity (SIDI, 1+), Shannon and Simpson's evenness (SHEI, SIEI, respectively, 0-1).

Year	CONT	SIDI	SHEI	SIEI
1972	26.6	1.60	0.90	0.90
1978	26.2	1.57	0.88	0.90
1982	21.0	1.67	0.93	0.95
1987	22.6	1.60	0.89	0.92
1992	22.3	1.63	0.91	0.94
2001	23.5	1.67	0.86	0.93

and NFBG within the ARI increased by 766.3 ha, whereas the area of the other cover types decreased by 760.1 ha (Table 7).

JP decreased the fastest within the ARI throughout the 29-year interval, losing approximately 13.3 ha yr^{-1} (~1.5% per year) (Table 7). The initial 14 years (1972 to 1986) represented the largest contributors to area change, with losses ranging between 44.1 and 48.2 ha yr^{-1}. NFBG increased the fastest within the ARI, gaining approximately 10.8 ha yr^{-1} (~ 1.2% per year). The rates of ARI change for the RFS, MH, MHC, and RP cover types fluctuated throughout the

Table 6. Edge characteristics of cover types by year. Given are total edge length (m), mean edge length (m), standard deviation of mean edge length (m) and edge density (ED, m ha^{-1}).

Cover Type	Total Edge (m)	Mean Edge (m)	Std Mean Edge	ED (m ha^{-1})
1972				
MH	1757367	1390	10040	152
RP	901968	716	3096	255
MHC	3053205	777	1955	314
JP	2082837	640	1129	346
RFS	2025153	666	1908	281
NFBG	362178	576	849	270
Total	10182708			
1978				
MH	2283363	1275	8412	179
RP	1124439	797	2258	250
MHC	3583989	561	722	381
JP	1408869	557	678	356
RFS	1801200	594	1614	222
NFBG	185592	539	696	277
Total	10387452			
1982				
MH	2425977	675	3440	283
RP	2017095	760	2442	313
MHC	3157401	776	2011	315
JP	1905339	386	372	468
RFS	2576343	668	2050	295
NFBG	487635	510	638	317
Total	12569790			
1987				
MH	2965596	715	4632	267
RP	1456236	609	1895	263
MHC	3888768	577	1216	307
JP	936624	351	509	466
RFS	1937430	455	782	357
NFBG	701898	573	956	270
Total	11886552			
1992				
MH	2402607	662	4386	220
RP	1667478	640	2182	279
MHC	3307938	718	2218	293
JP	984846	311	210	526
RFS	2068074	504	1171	319
NFGB	681549	615	1435	239
Total	11112492			
2001				
MH	2707044	737	2867	268
RP	1470942	505	1297	341
MHC	3723867	515	1126	386
JP	1089384	311	232	526
RFS	2908824	556	2930	314
NFGB	1414797	469	1403	357
Total	13314858			

Table 7. Area of road influence (ARI) change (ha), mean and percent rate of area change (ha yr^{-1} and % yr^{-1} respectively) of each cover type between classification years.Asterisk (*) indicates classification errors that are larger than the rates of change observed in each patch type.

Cover Type	ha	ha yr^{-1}	%/yr
Change: 1972 to 1978			
MH	41.4	6.9	0.9
RP	110.5	18.4	2.4
MHC	119.0	19.8	2.6
JP	− 264.8	− 44.1	− 5.8
RFS	106.6	17.8	2.3
NFBG	− 122.7	− 20.4	− 2.7
Change: 1978 to 1982			
MH	− 11.4	*-2.9	− 0.4
RP	258.4	64.6	8.2
MHC	− 125.9	− 31.5	− 4.0
JP	97.1	24.3	3.1
RFS	− 253.5	− 63.4	− 8.0
NFBG	46.1	11.5	1.5
Change: 1982 to 1987			
MH	205.3	41.1	3.8
RP	17.4	3.5	0.3
MHC	314.6	62.9	5.8
JP	− 240.8	− 48.2	− 4.4
RFS	− 276.3	− 55.3	− 5.1
NFBG+7+7+7	− 36.9	− 7.4	− 0.7
Change: 1987 to 1992			
MH	339.7	67.9	6.6
RP	16.7 *	− 3.3	− 0.3
MHC	− 143.3	− 28.7	− 2.8
JP	− 265.2	− 53.0	− 5.2
RFS	− 84.2	− 16.8	− 1.6
NFBG	174.0	34.8	3.4
Change: 1992 to 2001			
MH	− 125.3	− 13.9	− 0.9
RP	− 263.1	− 29.2	− 1.9
MHC	− 212.3	− 23.6	− 1.5
JP	− 159.4	− 17.7	− 1.2
RFS	514.3	57.1	3.7
NFBG	252.0	28.0	1.8
Change: Total			
MH	3.5 *	0.1	0.01
RP	106.5	3.7	0.4
MHC	− 48.0	− 1.7	− 0.2
JP	− 387.0	− 13.3	− 1.5
RFS	7.0 *	0.2	0.03
NFBG	312.5	10.8	1.2

entire time period, giving little indication of long term increasing or decreasing trends.

Discussion

Landscape composition

Composition and structure of the plantation pine systems (JP and RP) and the pine-barrens are typically quite different from pre-European pine systems. Historically (last 140 years), the pine-barrens and pine cover types have declined throughout northern Wisconsin, whereas there have been increases within the hardwood cover types (Radeloff et al. 1999a). Results from our classified images indicated a reverse trend to the historical accounts within all cover types except JP.

JP was the only cover type that followed the historical trend, showing not only decreases in overall area, but also a higher annual rate of change than other cover types. Typically JP is clearcut on a 40 to 60 year rotation; otherwise these systems would become fire hazards or transition into MHC. Since a majority of the JP was planted in the late 1930s to mid 1940s, most plantations were reaching harvestable age in the early and late 1970s, as indicated by the sharp decrease in JP between 1972 and 1982. This corresponds to the Forest Service stand records, which indicated that a large amount of JP was removed during the mid 70s and early 80s. Harvesting within the subsequent 19 years was likely due to; 1) stand maturity, 2) reducing the planting stock used throughout CNF in the 1930s because it was not site specific, coming from sources in Michigan and southern Wisconsin, and 3) large outbreaks of jack pine budworm (*Choristoneura pinus pinus*) and gall rust (*Endocrnartium harknessii*) that precipitated considerable salvage cutting between 1990 and 1995 (Radeloff et. al 1999b).

RP followed a different trend than JP due largely to the higher wood quality of the species. The stands planted by the CCC consisted primarily of RP due to its availability and economic value. Under the current management plan, the rotation age for RP is between 100 and 150 years, when either a shelterwood or seed tree cut is implemented on a harvestable stand (Forest Service 1986). Within the 150-year rotation, mature stands are intermittently thinned on a 7-15 year interval to allow release of dominant trees.

Prior to 1986 many of the RP plantations were thinned. Thinning within these mature stands was not easily distinguishable with the moderate-resolution images used in this study, and throughout most of this time period there were increases in RP within the images. An alternative sub-pixel classification might have resolved the discrepancy between the cover classification and the known management practices (Radeloff et al. 1999a,b; Adams et al. 1995).

Management during this period did not consider other uses of the forest when making management decisions that led to decreases in open areas (i.e., pine-barrens) and an increase in pine species (Table 4). Following the implementation of the 1986 Forest Plan, there were significant changes within the RP stands throughout the entire landscape. Within the pine-barrens region, many hectares of red pine, jack pine and hardwoods were harvested and burned to allow for restoration of open areas for wildlife and numerous vegetation types (Vora 1993). Changes in the management of pine-barrens are evident by the increase maximum patch size, standard deviation of mean patch size and coefficient of variation during the last 14-year interval.

Many RP plantations approached harvestable age within the 1990s and this, combined with the synchronous increase in stumpage prices, restoration of the pine-barrens, and significant windthrow events resulted in an increased harvest within this cover type (Crow et al. 1999). The windthrow event that occurred in the northern pine plantations between 1992 and 2001 primarily affected the RP. The combination of all these factors resulted in a large decrease of RP from 1992 to 2001 (Table 4), in spite of the overall increase in the RP cover type from 1972 to 2001.

MH was the most continuous and dominant cover type in the 1970s, encompassing approximately 30% of the total landscape area. However, a significant decrease occurred between 1978 and 1982. Parallel but opposite changes (i.e., increases) in the area covered by the RFS and NFBG cover types indicated that large portions of the MH were removed during this period, partially due to the occurrence of a severe wind storm that felled many hectares of hardwood species. Subsequent to the windthrow event, salvage operations accounted for much of the loss of MH during this period. Following the decrease in 1982, the MH never fully regained its previous dominance, which may have resulted partially from the increased management focus on the pine-barrens where large portions of forested land were being converted into RFS in an effort to restore large areas to the historical pine-barrens ecosystem type.

Changes in MHC distribution could not be related to any specific management event or natural disturbance because it is not typically managed as a whole entity. Information gathered from the forest stand data indicated that instead of managing the stand as mixed forest it is separated into small patches and managed as either hardwood or conifer. This suggests that the hardwood and conifer species within the MHC forests have an aggregated rather then random distribution throughout our classified images. This corresponds with many RFS patches, in which there are islands of mature white pine within a matrix of regenerating young hardwoods and vice-versa.

The recent increase in the RFS cover type was primarily due to expansion of the pine-barrens located in the south central portion of the study area. Historically this area burned regularly, limiting successional development and maintaining a relatively open-canopy ecosystem that occupied a large portion of the pre-settlement landscape (about 15,400 ha within the CNF alone, Curtis 1959). By the late 1980s and early 1990s, almost 80% of this ecosystem had been converted to closed-canopy forests as a result of fire suppression and tree planting activities (USDA 1993). The shift to ecosystem management with its accompanying concerns for non-economic ecosystem values and functions, however, coincided with an effort on the part of the forest managers to restore large areas to pine-barrens using clearcutting and prescribed burning (Vora 1993; USDA 1993). This restoration effort shows clearly in our results (Figure 3) as a steady increase of RFS and NFBG in the pine-barrens area between 1987 and 2001. Current management relies largely upon prescribed burning to inhibit succession and maintain the open nature of the pine-barrens. Increases in the RFS typically occurred in conjunction with decreases in forested cover. For example, there was a large decrease in MH and an increase in RFS from 1978 to 1982. This suggests that harvesting activities during this time period cleared more forests than grew into the forested class.

Landscape structure and pattern

All cover types within our study site have become increasingly fragmented (Figure 5b,c, d) during the 29-year period. JP did not follow this trend precisely because of the substantial reductions in its area throughout the landscape. Fragmentation within the JP cover type has occurred because of changes in management direction and natural disturbance. The large amount of high-contrast, forest-clearcut edge that is created combined with the rapid decline in the extent of interior forest habitat is the primary criticism of dispersed-cutting patterns (Wallin et al. 1994). Changes in the microclimatic conditions along the forest-clearcut edge influence seedling establishment, growth, mortality, and competition interactions between plants, and subsequently alter structure, composition, and pattern (Chen et al. 1992; Saunders et al. 1991). High-contrast harvests combined with road density and reduction in fire occurrences has severely fragmented the JP cover type throughout the 29-year interval.

Prior to European settlement, fire was the primary stand-replacing disturbance in most forested ecosystems (Agee 1993), and for JP it is typically required for regeneration. Fire suppression within JP stands has increased the need for intensive management. Past management of JP primarily involved clearcutting, which did not fully mimic the fire process. Despite small-scale structural similarities between clearcutting and stand replacing fires, these disturbances differ in many ways (Cohen et al. 2002). For example, wildfires leave large patches of old forest intact (Van Wagner 1978), whereas clearcutting leaves no remnant forests older than rotation age. Additionally, patch aggregates created through harvesting have been typically smaller and more fragmented than those created by fire events, creating a “swiss-cheese” landscape pattern in which small, discrete patches are created and dispersed throughout the landscape. This exacerbates the problem of edge effects, since more edge exists where landscape patches are small and dispersed rather than aggregated. The JP exhibited this trend, with an increase in total edge length and a decrease in mean patch size.

Current management approaches within the JP stands implement a combination of clearcutting and prescribed burning in order to further mimic the fire processes in these stands and create opportunities for JP regeneration. Crow et al. (1999) found that reintroducing fire and using large harvest units on outwash moraines retained the coarse-grained heterogeneity that existed in these types of landscapes. Even so, severity of prescribed burns often differs from those of historical fires, impacting species, stands, and landscape patterns and processes (Cohen et al. 2002).

Patch pattern throughout the plantations in the northeast portion of the forest depend primarily on the harvesting regimes dictated by the Forest Service. RP has become increasingly fragmented within the last 29 years. Prior to 1986, management impacted patterns of fragmentation within the RP cover type, as seen by the large fluctuations in mean patch size (MPS) and edge length between 1972 and 1987 (Figure 5d). Following the change in management direction in 1986, there has been a continual trend in which the total area and mean patch size decreased and total edge length increased, suggesting that fragmentation and timber harvesting increased within this cover type. This corresponds with the increase in stumpage prices, restoration of pine-barrens, timber maturation, and windthrow events that occurred in the 1990s.

Although MH remained one of the most continuous cover types, its structure has changed substantially since the 1970s, with the primary causes being, again, windthrow events and harvesting. Patch sizes were significantly larger than any other cover type within any given year, whereas total edge length was relatively low and mean edge length relatively large, indicating limited fragmentation of this patch type prior to the 1980s. The most significant changes (i.e., decreases in mean patch size and area of MH within the ARI) occurred in 1982 and were related to the windthrow event that occurred in 1979. This indicates that, prior to 1986, natural disturbances had a greater impact than management on fragmentation within the hardwood cover type. Following 1986, reductions in patch size and increasing edge lengths could be related directly to the restoration of the pine-barrens, which resulted in the harvesting and burning of considerable areas of the hardwood cover type.

The variety of management objectives implemented in the MHC cover type influences patterns of fragmentation and species composition. Because management often takes place on small clumps of either conifers or hardwoods within the stand and depends on composition, which varies within this cover type throughout the landscape, little could be concluded regarding patch size and edge characteristics of the MHC. Mixtures of hardwoods and conifers often occur in older stands where hardwood species have grown up around remnant pines. Additionally, thinning in RP plantations or senescence in JP creates canopy openings that can be filled by regenerating hardwood species such as red maple or northern pin oak (*Quercus ellipsoidalis*). Clearly, management activities may differ considerably between these two types of stands, obscuring structural patterns related to one type or the other.

Although currently the pine-barrens are the predominant area containing RFS and NFBG, these patches also exist throughout the landscape in the ARI and in areas of recent disturbance (e.g., recent harvesting activities). Because harvested areas experiencing regrowth were included in the RFS cover class, the distribution of this cover type was typically patchy and fragmented, as shown by the small patch sizes and large edge length.

The landscape metrics (e.g., Shannon and Simpson's evenness, contagion and Simpson's diversity) did not experience large changes between 1972 and 2001. Although only small changes occurred, inferences can be made about landscape pattern and structure during the 29-year interval. Evenness measures the distribution of area among the different patch types across the landscape. Shannon and Simpson's evenness values were high, which suggests an uneven distribution of patch types. Contagion measures the extent of patch aggregation. Contagion values were relatively low, which is characteristic of landscapes with small, dispersed patches. Across the 29-years, the contagion value decreased, which indicates a decrease in aggregation of each cover type. Wolter et al. (2002) found similar results and concluded that increased evenness and decreased contagion values suggest increasing fragmentation of contiguous, mature forest types. In our landscape the most contiguous patch type (MH) experienced the greatest fragmentation, which directly corresponds to the greatest change in evenness and contagion.

Landscape structure within our study site corresponded to the fragmentation syndrome outlined by Baker (2000). Landscape fragmentation has increased in the study area since 1972, following a trend common throughout the forests of the eastern United States and elsewhere (Vogelmann 1995; Smith et al. 1993; Cohen et al. 2002). Fragmentation in the CNF is caused by an extensive road network and forest management activities, primarily timber harvesting.

Area of Road Influence (ARI)

Roads are unique in their impacts due to their persistence throughout the surrounding landscape, and they are typically the primary cause of forest fragmentation (Saunders et al. 2002). Within our study site the RP plantations, JP, and open areas are typically the most densely roaded areas which accounts largely for the smaller patch sizes and longer edge lengths in these cover types. ARI within the Chequamegon landscape impacted frequency and distribution of each cover type differently. Our results support those found by Saunders et. al. (2002). Saunders found that open areas, red, white and jack pine plantations are the most densely roaded areas, which coincides with our results of lower hardwood distribution in the ARI as compared to the entire landscape (Table 3). High road density in the pine type is due to the amount of harvesting done within these areas.

The amount of JP within the interior patches and ARI decreased throughout the 29-year interval. The frequency of distribution within both areas was relatively similar for this cover type, indicating that road systems did not significantly affect JP composition. The only year that had a large difference between the proportion of JP within ARI and within interior patches was 1992, which could be due to the location of insect and disease epicenters or roads that could be corridors for insect dispersal. The area covered by RP within the interior patches and within the ARI increased throughout the 29-year interval. The frequency of distribution within both areas was relatively similar for this cover type, indicating that road systems did not significantly affect RP representation.

Frequency of distribution within the MH was smaller in the ARI as compared to the interior forest matrix (Table 3). This is due to a combination of low road density (typically higher in pine types) and high disturbance (i.e., mowing) for this cover type. As fragmentation within this class increased, the difference between frequency of distribution within the ARI and interior patches decreased. For example, in 1972 when there were large homogeneous patches of hardwoods, the difference between hardwood representation in the ARI and in interior patches was ~ 48%, whereas in 2001 there was ~ 42% difference.

The frequency of MHC along the roadsides was similar to the interior patches suggesting some difficulties in assessing the impacts of the ARI on this cover type. There is little indication if MHC is harvested heavily or if it is left for long periods similar to that of the MH. Further documentation and understanding of this cover type and the associated management objectives is needed before any relevant conclusions can be made.

RFS and NFBG had a higher frequency of distribution along the ARI as compared to the interior forest matrix. Most disturbances within this landscape are elated to management. To increase efficiency and profitability of harvests most cut are along the road corridor, which leads to an increase in younger and non-forested stands along the ARI. Because roads and roadsides constitute disturbance corridors that often contain harsher environmental conditions and more frequent disturbance (e.g., mowing) than areas away from roads, it was expected that RFS would comprise a relatively large proportion of the ARI.

Since our study focused on the main road system this limited inferences of road effects within our study area. Further information, such as road type, use, management (i.e., maintenance), traffic, etc., and road dynamics (i.e., obliteration and construction) would be necessary to develop a full understanding of ARI in the landscape.

Conclusions

Changes in management objectives and natural disturbances have had clear influences on landscape patterns and composition in the CNF throughout the past 29 years. Windthrow events, disease outbreaks, and changes in stumpage value greatly influenced both composition and structure within our study site. In particular, forested cover types tended to decrease and open-canopy (RFS, NFBG) cover types tended to increase as a result of these disturbances.

As forest management is also constantly changing, in response to cultural and economic changes, it is important to take an adaptive, broad-scale, and long-term approach to management, building into the process a way to evaluate the multi-scale responses of the landscape to management activities over time and make alterations if necessary. Remote sensing and GIS provide important tools for making these evaluations and helping to assess the link between management activities and ecological consequences. The revision process for the Chequamegon Forest Plan has been underway since 1996. It will be informative to track the future changes in management and the corresponding changes in landscape structure, especially since broad-scale management is becoming more common. Once patterns are known, processes can be inferred future studies should focus on furthering our knowledge of the links between broad-scale pattern and ecological, social, and economic processes in addition to identifying changes in patterns so that forest managers are better able to deal effectively with the often conflicting demands society places on its public forests.

Acknowledgements

This study is partially supported by the National Science Foundation (DEB 0129405), the Northern Global Change Program of the USDA Forest Service and the Joint Fire Science Program. Phil Freeman and Robert Raade supported with descriptions and information on past and current management objectives along with the compilation of the database from the Washburn District of the CNF. Mark Theisen assisted in describing disturbances as well as silvicultural techniques used during the past 29-years; he is located in Chequamegon-Nicolet National Forest, Rhinelander, WI. David Campbell provided additional information on road systems within the forest during the past 29 years. Deb Neher, Soung Ryoul-Ryu and Daolan Zheng from the University of Toledo EEES Department added methodological and conceptual support. Kevin Czajkowski and Karl Schneider from the University of Toledo Geography and Planning Department added technical support. Amy Campbell assisted in compiling the GPS database.

References

Adams J.B., Sabol D.E., Kapos V., Filho R.A., Roberts D.A., Smith M.O. and Gillespie A.R. 1995. Classification of multispectral images based on fractions of endmembers: Application to land-cover change in the Brazilian Amazon. Remote Sensing of Environment 52: 13–154.

Agee J.K. 1993. Fire ecology of Pacific Northwest forests. Island Press, Washington, DC, USA.

Albert D.A. 1995. Regional landscape ecosystems of Michigan, Minnesota, and Wisconsin: a working map and classification. USDA FS Gen. Tech. Rep. NC-178. St. Paul, Minnesota, USA.

Bailey R.G. 1995. Description of the ecoregions of the United States (2nd ed.). Misc. Publ. No. 1391, USDA Forest Service, Washington, DC, USA.

Brosofske K.D., Chen J., Crow T.R. and Saunders S.C. 1999. Vegetation responses to landscape structure at multiple scales across northern Wisconsin, USA, Pine-barrens landscape. Plant Ecology 143: 203–218.

Brosofske K.D., Chen J. and Crow T.R. 2001. Understory vegetation and site factors: Implication for a managed Wisconsin landscape. Forest Ecology and Management 146: 75–87.

Chen J., Franklin J.F. and Spies T.A. 1992. Vegetation response to edge environments in old-growth Douglas-fir forests. Ecological Applications 2: 387–396.

Chen J., Saunders S.C., Crow T.R., Naiman R.J., Brosofske K.D., Mroz G.D., Brookshire B.L. and Franklin J.F. 1999. Microclimate in forest ecosystems and landscape ecology. BioScience 49: 288–297.

Cohen W.B., Spies T.A. and Fiorella M. 1995. Estimating the age and structures if forest in a multi-ownership landscape of Westerns Oregon, USA. International Remote Sensing of Environment 16(4): 721–746.

Cohen W.B., Spies T.A., Alig R.J., Oetter D.R., Maiersperger T.K. and Fiorella M. 2002. Characterizing 23 years (1972-95) of stand replacement disturbance in Western Oregon forests with Landsat imagery. Ecosystems 5: 122–137.

Crow T.R., Host G.E. and Mladenoff D.J. 1999. Ownership and ecosystem as sources of spatial heterogeneity in a forested landscape, Wisconsin USA. Landscape Ecology 14: 449–463.

Cushman S.A. and Wallin D.O. 2000. Rates and pattern of change in the central Sikhote-alin Mountains, Russia Far East. Landscape Ecology 15: 643–659.

Curtis J.T. 1959. The Vegetation of Wisconsin. University of Wisconsin Press, Madison, Minnesota, USA.

Duda R.O. and Hart P.E. 1973. Pattern Classification and Scene Analysis. John Wiley and Sons, New York, New York, USA, pp. 100–125.

Euskirchen E.S., Chen J. and Bi R.C. 2001. Effects of edges on plant communities in a managed landscape in northern Wisconsin. Forest Ecology and Management 148: 93–108.

Franklin J.F. and Forman R.T.T. 1987. Creating landscape pattern by forest cutting: ecological consequences and principles. Landscape Ecology 1: 1–5.

Heinselman M.L. 1981. Fire intensity and frequency as factors in the distribution and structure of northern ecosystems. In: Mooney H.A., Bonnicksen T.M., Christensen N.L. et al. (eds), Fire regimes and ecosystem properties: Proceedings of the conference; 1978 December 11-15; Honolulu, HI. Gen. Tech. Rep. WO-26. Department of Agriculture, Forest Service, Washington, DC, USA, pp. 7-57.

Jakubauskas M.E. 1996. Thematic Mapper characterization of lodgepole pine seral stages in Yellowstone National Park, USA. Remote Sensing of Environment 56: 118–132.

McNab W.H. and Avers P.E. 1994. Ecological subregions of the United States: Section descriptions. Administrative Publication WO-WSA-5. USDA Forest Service, Washington, DC, USA.

Mas J.F. 1999. Monitoring land-cover changes: a comparison of change detection techniques. International Journal of Remote Sensing 20: 139–152.

McGarigal K., Cushman S.A., Neel M.C. and Ene E. 2002. FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps. Computer software program produced by the authors at the University of Massachusetts, Amherst. Available at the following web site: www.umass.edu/landeco/research/fragstats/fragstats.html

Overbay J.C. 1992. Ecosystem management – keynote address. Proceedings of the National Workshop: Taking an Ecological Approach to Management, WO-WSA-3. USDA Forest Service, Washington, DC, USA, pp. 3–22.

Pan D., Domon G., Marceau D. and Bouchard A. 2001. Spatial patterns of coniferous and deciduous forest patches in an Eastern North America agricultural landscape: the influence of land use and physical attributes. Landscape Ecology 16: 99–110.

Radeloff V.C., Mladenoff D.J. and Boyce M. 1999a. Effects of jack pile budworm defoliations and salvage harvesting on landscape patterns. Ecological Applications 10: 233–247.

Radeloff V.C., Mladenoff D.J. and Boyce M. 1999b. Detecting jack pine budworm defoliation using spectral mixture analysis: Separating effects from determinants. Remote Sensing of Environment 69: 156–169.

Saunders D.A., Hobbs R.J. and Margules C.R. 1991. Biological consequences of ecosystem fragmentation: a review. Conservation Biology 5: 18–32.

Saunders S.C., Chen J., Crow T.R. and Brosofske K.D. 1998. Hierarchical relationships between landscape structure and temper-

ature in a managed forest landscape. Landscape Ecology 13: 381–395.
Saunders S.C., Chen J., Drummer T.D. and Crow T.R. 1999. Modeling temperature across edges over time in a managed landscape. Forest Ecology and Management 117: 17–31.
Saunders S.C., Mislivets M.R., Chen J. and Cleland D.T. 2002. Effects of roads on landscape structure within nested ecological units of the Northern Great Lakes Region, USA. Biological Conservation 103: 209–225.
Schowengerdt R.A. 1983. Techniques for image processing and classification in remote sensing. Academic Press, New York, New York, USA.
Smith B.E., Marks P.L. and Gardescu S. 1993. Two Hundred Years of Forest Cover Changes in Tompkins County, New York. Bulletin of the Torrey Botanical Club 120: 229–247.
USDA Forest Service 1986. Chequamegon Forest Plan. Washburn Ranger District, Chequamegon National Forest, Washburn, Wisconsin, USA.
USDA Forest Service 1993. Landscape Level Analysis: Desired Future Vegetative Condition. Washburn Ranger District, Chequamegon National Forest, Washburn, Wisconsin, USA.
Van Wagner C.E. 1978. Age-class distribution and forest fire cycle. Canadian Journal of Forest Resources 8: 220–227.
Vogelmann J.E. 1995. Assessment of Forest Fragmentation in Southern New England Using Remote Sensing and Geographic Information Systems Technology. Conservation Biology 9: 439–449.
Vora R.S. 1993. Moquah Barrens: Pine-barrens restoration experiment initiated in Chequamegon National Forest. Restoration and Management Notes 11: 39–44.
Wallin D.O., Swanson F.J. and Marks B. 1994. Landscape pattern response to changes in pattern generation rules: Land-use legacies in forestry. Ecological Applications 4(3): 569–580.
Walsh S.J., Butler D.R. and Malanson G.P. 1998. An overview of spatial scale, pattern, and process relationships in geomorphology: a remote sensing and GIS perspective. Geomorphology 21: 183–205.
Watkins R.Z., Chen J., Pickens J. and Brosofske K.D. 2003. Effects of forest roads on understory plants in a managed hardwood landscape. Conservation Biology 17: 411–419.
Wolter P.T. and White M.A. 2002. Recent forest cover type transitions and landscape structural changes in northeast Minnesota. Landscape Ecology, 17: 133–155.
Zheng D. and Chen J. 2001. Edge effects in fragmented landscapes: a generic model for delineating area of edge influences (D-AEI). Ecological Modeling 132: 175–190.
Zheng D., Wallin D.O. and Hao H. 1997. Detecting rates and patterns of landscape change in the Changhai Mountain area between 1972-1988 using satellite data. Landscape Ecology 12: 241–254.